Cattle feed or bioenergy? Consequential life cycle assessment of biogas feedstock options on dairy farms

Styles, D.; Gibbons, J.; Williams, A.P.; Stichnothe, H.; Chadwick, D.R.; Healey, J.R.

Global Change Biology Bioenergy

DOI:
10.1111/gcbb.12189

Published: 10/08/2015

Cattle feed or bioenergy? Consequential life cycle assessment of biogas feedstock options on dairy farms

David Styles[†], James Gibbons[†], Arwel Prysor Williams[†], Heinz Stichnothe[‡], David Robert Chadwick[†], John Robert Healey[†]

[†]School of Environment, Natural Resources & Geography, Bangor University, LL57 2UW, Wales

[‡]Thünen Institute, Bundesallee 50, 38116 Braunschweig, Germany.

***Corresponding author: Email: d.styles@bangor.ac.uk Tel.: (+44) (0) 1248 38 2502**

Abstract

On-farm anaerobic digestion (AD) of wastes and crops can potentially avoid greenhouse gas (GHG) emissions, but incurs extensive environmental effects via carbon and nitrogen cycles and substitution of multiple processes within and outside farm system boundaries. Farm models were combined with consequential life cycle assessment (CLCA) to assess plausible biogas and miscanthus heating pellet scenarios on dairy farms. On the large dairy farm, the introduction of slurry-only AD led to reductions in global warming potential (GWP) and resource depletion burdens of 14% and 67%, respectively, but eutrophication and acidification burden increases of 9% and 10%, respectively, assuming open tank digestate storage. Marginal GWP burdens per Mg dry matter (DM) feedstock co-digested with slurry ranged from –637 kg CO_2e for food waste to +509 kg CO_2e for maize. Co-digestion of grass and maize led to increased imports of concentrate feed to the farm, negating the GWP benefits of grid-electricity substitution. Attributing grass-to-arable land use change (LUC) to marginal wheat feed production led to net GWP burdens exceeding 900 kg CO_2e Mg^{-1} maize DM co-digested. Converting the medium-sized dairy farm to a beef-plus-AD farm led to a minor reduction in GWP when grass-to-arable LUC was excluded, but a 38% GWP increase when such LUC was attributed to marginal maize and wheat feed required for intensive compensatory milk production. If marginal animal feed is derived from soybeans cultivated on recently converted cropland in South America, the net GWP burden increases to 4099 kg CO_2e Mg^{-1} maize DM co-digested – equivalent to 55 Mg CO_2e yr^{-1} per hectare used for AD-maize cultivation. We conclude that AD of slurry and food waste on dairy farms is an effective GHG mitigation option, but that the quantity of co-digested crops should be strictly limited to avoid potentially large international carbon leakage via animal feed displacement.

Keywords: anaerobic digestion; LCA; farm models; mitigation; GHG; resource efficiency; eutrophication; land use change; greenhouse gas

Introduction

Drivers for farm scale anaerobic digestion

Livestock agriculture is responsible for 10% of anthropogenic GHG emissions across Europe (Westhoek et al., 2011), and the wider agricultural sector is responsible for 94% of ammonia (NH_3) emissions that contribute to acidification and eutrophication (EEA, 2012). NH_3 emissions originate largely from manure storage and land application. In the UK alone, 31 Gg yr^{-1} of NH_3 arise from manure storage and a further 54.2 Gg yr^{-1} from manure application to soils (Misselbrook et al., 2012). Farm-scale anaerobic digestion (AD) is one option that can reduce GHG emissions, and potentially NH_3 emissions, from manure storage whilst also capturing biogas to run combined heat and power (CHP) boilers, displacing fossil energy carriers and associated GHG emissions. In addition, AD increases total ammonical N (TAN) of feedstock manures, enhancing their fertilizer replacement value and potentially improving N-use efficiency (Holm-Nieselson et al., 2009), but also potentially increasing NH_3 emissions during digestate storage and spreading (Rehl&Müller, 2011).

Farm-scale AD is an established technology in countries such as India, China, Germany, Austria and Switzerland. There are estimated to be over 7 000 AD plants in Germany, partially fed by AD-purpose-grown crops (AD-PGCs) such as maize grown on up to 750 000 ha of land (FNR, 2013). Within the UK, as of early 2013, there were only 45 AD plants using agricultural feedstocks and 48 using food waste (ADBA, 2013) but there is an increasing momentum behind implementation with over 100 plants under construction or planned as of 2013 (ADBA, 2013). Economic drivers for farm AD include historically high fertilizer and energy prices and feed-in-tariffs of up to £0.15 per kWh for biogas electricity (FIT Ltd, 2013). An evidence summary of AD in the UK compiled by Defra (2011) concluded that economic drivers would encourage the incorporation of 30-40% AD-PGC into manure-fed AD units on dairy farms. Meanwhile, Mistry et al. (2011a; b) showed that inclusion of food and other wastes as AD feedstock improves the economic viability of AD plants, and could lead to annual GHG mitigation in the UK of over 3 Pg CO_2e, partly attributable to avoided landfill emissions, compared with economically viable annual GHG mitigation of 63 Gg CO_2e using only farm wastes for AD feedstock. However, that study focussed on the geographic distribution of AD feedstocks, and based its estimates of environmental effects on highly simplified assumptions such as 1% leakage of CH_4 during digestion and storage and no land use change (LUC) from AD-PGC.

Consequential life cycle assessment of bioenergy

Life cycle assessment (LCA) is a systems approach used to quantify material and energy flows, and associated environmental burdens, arising over the production, consumption and disposal or recycling of a specified quantity (functional unit) of product or service (ISO, 2006a;b). Environmental impact categories particularly relevant to agricultural systems include global warming potential (GWP), eutrophication potential (EP), acidification potential (AP) and resource depletion potential (RDP).

If land is used to grow bioenergy feedstocks, or if food waste is diverted from existing waste management to farm AD units, complete assessment of global resource and environmental effects requires an expansion of LCA boundaries beyond the farm system. Whilst attributional LCA (ALCA) assesses environmental effects directly attributable to the system delivering the primary functional unit of interest, consequential LCA (CLCA) expands system boundaries to account for marginal economically-induced effects of system modifications throughout the wider economy (Weidema, 2001). Consequential LCA is increasingly being applied to assess bioenergy interventions in agricultural and energy systems (e.g. Dalgaard et al., 2008; Thomassen et al., 2008; Mathieson et al., 2009; Dandres et al., 2011; DeVries et al., 2012; Hamelin et al., 2012; Rehl et al., 2012; Steubing et al., 2012; Hamelin et al., 2014).

Marginal effects of displaced food production in agricultural CLCA are particularly difficult to estimate, usually comprising a mix of intensification and land transformation across national boundaries, with possible cascading displacement of crops (Schmidt, 2008; Kløverpris et al., 2008; Mulligan et al., 2010). Such effects may be estimated based on market data or predicted using general equilibrium economic modelling (Earles et al., 2012; Marvuglia et al., 2013), involving large uncertainties around price trends, future policy interventions, elasticities of demand and marginal technologies, amongst other factors (Schmidt, 2008). Zamagni et al. (2012) argue that CLCA is not well systemised, and may generate misleading evidence for policy makers where complex methods are not adequately described. Scmidt (2008), Marvuglia et al. (2013) and Vazquez-Rowe et al. (2014) demonstrate how simplified, qualitative scenarios defined by expert judgement can be used to estimate marginal effects in CLCA. Estimating such effects at the global scale is very important given that almost 60% of land required for EU consumption is located outside of the EU (Tukker et al., 2013).

Environmental performance of farm biogas

In terms of farm environmental balance, N-availability benefits of AD (Holm-Nieselson et al., 2009) are opposed by increased NH_3 emissions from digestate storage and spreading, compared with non-digested manures (Rehl&Müller, 2011). Hamelin et al. (2011) and Boulamante et al. (2013) found that biogas produced from manure resulted in large GHG emission reductions compared with the counterfactual of manure storage, largely through avoided manure storage CH_4 emissions and grid electricity displacement. Buolamante et al. (2013) emphasised the importance of closed digestate storage tanks to improve environmental performance, principally by reducing NH_3 emissions and thus the EP and AP burdens from the system. Unit process data from commercial LCA databases and IPCC (2006) GHG emission calculations fail to fully capture resource flow and N cycling effects of interventions such as AD. Del Prado et al. (2013) demonstrate the importance of farm models to accurately quantify net effects of agricultural technologies and management practices. The study presented here involved the development of detail farm models to capture AD effects.

Tonini et al. (2012) showed that indirect land use change (iLUC) may cancel the GHG savings associated with bioenergy systems based on grass, willow, and miscanthus feedstocks, whilst Hamelin et al. (2014) showed that iLUC led to net GHG emission increases when crops are used as co-substrate for AD of manure. Tufvesson et al. (2013) showed that the calculated GHG mitigation potential of AD using industrial residues is considerably reduced when system expansion is applied to consider alternative use of residues such as animal feed. Meanwhile, European demand for soy bean meal extract (SBME) has been implicated as a driver for LUC in Brazil and Argentina (Hortenhuber et al., 2011), and incurs additional effects such as palm oil substitution by soy oil co-produced with SBME (Dalgaard et al., 2008; Thomassen et al., 2008). Digesting crops in farm AD units could lead to various marginal effects, including iLUC incurred by displaced fodder production (Hamelin et al., 2014), or iLUC incurred by marginal increases in demand for concentrate feeds such as winter wheat and SBME. As far as these authors are aware, the latter pathway of iLUC driven by farm bioenergy has not been evaluated at the farm scale in the scientific literature. This paper attempts to fill that gap.

Aims and objectives

The primary purpose of this paper is to assess the net environmental effects of bioenergy options on dairy farms, based on farm economic and environmental modelling combined with CLCA. Multiple data sets were integrated within a scenario tool ("LCAD") developed in MS Excel to inform policy makers and prospective farm AD operators. The LCAD tool was commissioned by the UK government's department of agriculture and the environment to compare the environmental performance of AD and farm bioenergy options (Defra, 2014).

The methods section of this paper describes the main elements of the LCAD tool, and the results section presents a selection of outputs from the tool to: (i) quantify the net environmental effects of plausible bioenergy scenarios on dairy farms; (ii) assess the influence of AD design and management factors on environmental performance; (iii) compare the environmental efficiency of bioenergy feedstocks on dairy farms, accounting for animal feed displacement effects.

Materials and Methods

Scope and boundaries

The primary function of the LCAD tool is to compare the net environmental effects of plausible dairy farm bioenergy options for representative medium and large UK dairy farms, and to provide insight into key drivers and uncertainties around these effects. Reference systems are representative medium and large UK dairy farms, and the functional unit was taken to be annual food production on the baseline farm: 4 149 102 L milk and 9 242 kg live weight beef output for the large dairy farm, and 1 013 548 L milk and 2 446 kg live weight beef output for the medium dairy farm. Four environmental impact categories were accounted for based on CML (2010) characterisation factors (Table 1). Infrastructure was excluded from the scope. The functional unit relates to one year of farm operation. The temporal scope is approximately 10 years, considering the time required for wider adoption of farm bioenergy options and enabling the consideration of current prevailing technologies for counterfactual processes, such as composting for waste management. Farm modelling was based on current yield and economic data. (Insert Table 1 about here).

Results are expressed as net annual environmental burden changes for bioenergy scenarios compared with reference (baseline) farms, maintaining constant food production, applying CLCA to consider processes avoided or incurred outside the farm system boundary (Figure 1). These processes include avoided marginal grid-electricity generation, avoided marginal heat production, avoided food waste disposal, and avoided marginal animal feed (hay and concentrate) production. LCAD users choose from short-listed farm and marginal processes specified in Table 2 to create simplified narratives. This links specific management decisions and marginal consequences to environmental effects, and bounds these effects for dairy farm bioenergy options, using a scenario approach similar to Schmidt (2008), Marvuglia et al. (2013) and Vazquez-Rowe et al. (2014). (Insert Figure 1 and Table 2 about here).

LCAD tool and scenario development

The process of LCAD tool development involved significant input from a technical working group (TWG) comprising AD consultants, suppliers and operators, officials from Defra and DECC government agencies, and representatives from agricultural organisations including the National Farmers' Union. The development process was as follows: (i) baseline dairy farm models were parameterised based on UK farm statistics (FBS, 2013) and farm optimisation using the Farm-adapt model (Gibbons et al., 2006); (ii) agronomically and economically plausible farm bioenergy scenarios were defined through consultation with the TWG (TWG, 2013) and parameterised in Farm-adapt; (iii) detailed LCA was undertaken for baseline and scenario farms; (iv) CLCA was undertaken for processes avoided or incurred outside the farm system boundary in the bioenergy scenarios, including those listed in Table 2; (v) LCA output sheets were linked to a user interface in MS Excel where users select scenarios and scenario permutations based on key variables listed in Table 2, providing for sensitivity analyses; (vi) corrections and refinements were made to the LCAD tool following review by the TWG. The LCAD tool is available online (Defra, 2014).

Farm model

Baseline medium and large dairy farms (Table S1.1) were originally parameterised in Defra (2009b), derived from representative farm typologies informed by UK Farm

Business Survey statistics. The Farm-adapt model is described in Gibbons et al. (2006), and optimises farm operations to maximise net margin according to input and output prices. Economic data were obtained from Nix (2010), DairyCo (2013a) and Defra (2012). All animal excreta generated indoors on the two farms is assumed to be stored in liquid slurry tanks with crust covers under default LCAD tool settings, with lagoon storage modelled as an alternative baseline permutation.

Crop nutrient requirements (Table S1.2 were based on UK fertiliser recommendations (RB209) for high grass growth on good quality soils with soil P and K indices of 2 (Defra, 2010). Mineral fertiliser application rates were calculated by subtracting from the crop nutrient requirements: (i) the plant-available nutrients delivered by slurry or digestate applications, as determined by runs of the MANNER-NPK model (Nicholson et al., 2013) parameterised for the time, quantity and method of application and slurry N fractionation as described below; (ii) soil N supply as estimated from the soil type and previous crop management (Defra, 2010). Grassland re-seeding every five years incurs annualised residue-N input, tractor diesel consumption and seed production burdens. Soil organic carbon content under grassland and maize areas was assumed to be in equilibrium on baseline farms.

Direct emission factors applied to on- and off-farm processes are summarised in Table 3. Enteric CH_4 emissions were calculated from equation 10.21 of IPCC (2006), based on feed intake and using a methane conversion factor (Ym) value of 6.5%. Manure management CH_4 emissions were calculated from equations 10.23 and 10.24 of IPCC (2006), based on animal feed intake calculated in Farm-adapt and an average digestibility factor of 75% (Brown et al., 2012). Housing emissions of NH_3 were calculated per livestock unit (LU) per day for housed dairy cows and calves following

Misselbrook et al. (2012). The volume of manure generated and total N excretion (N_{ex}) indoors and outdoors were calculated in using the IPCC (2006) Tier 2 approach from feed intake, gross energy (GE) requirements and the proportion of time animals were housed (Table S1.1). Slurry storage NH_3-N EFs of 0.05 and 0.515 total ammonical N (TAN) for tanks (crusted) and lagoons were taken from Misselbrook et al. (2012), assuming 60% of N_{ex} is TAN (Webb and Misselbrook, 2004). (Insert Table 3 about here).

Field losses of NH_3 and NO_3^- from slurry and digestate application were calculated using MANNER-NPK (Nicholson et al., 2013), run under the following inputs: post-code near Exeter in SW England; moist sandy-clay-loam soil and subsoil; moderate breeze with no rain in the subsequent six hours; broadcast (splash-plate) spreader for baseline slurry application and a trailing shoe spreader for liquid digestate application. It was assumed that 10% of fertiliser, residue and grazing N inputs leach to water (Duffy et al., 2013). An NH_3-N EF of 0.06 was applied to grazing TAN deposition (Misselbrook et al., 2012). Indirect N_2O-N emissions were calculated as per IPCC (2006): 0.01 of volatilised N, following deposition, and 0.0075 of leached N. Phosphorus losses were estimated at 0.03 of all P amendments to surface soils (Johnes et al., 1996; Withers, pers. comm. 2013). For in-field diesel combustion in tractors, NO_x emissions were approximated to EURO III emission standards for 75-130 kW off-road vehicles assuming 30% engine efficiency (Dieselnet, 2013). Emissions of SO_x calculated from 10 mg S per kg red diesel (DfT, 2010) were trivial and therefore disregarded.

Diesel consumption for field operations was modelled in Farm-adapt (Gibbons et al., 2006), farm electricity consumption was calculated as 350 kWh per milking cow per year (Warwick HRI, 2007) and farm heating-oil consumption estimated at 50% of electricity use. Farmhouse heat demand of 40 000 kWh yr^{-1} was calculated as the average specific heat demand for a UK home (160 kWh m^{-2} yr^{-1}) multiplied by a floor area of 250 m^2, and was assumed to be supplied by an oil boiler. This is not included in the baseline farm LCA, but is assumed to be displaced by biogas heat in the farm AD scenarios. Upstream environmental burdens for inputs to the farm (Table 4) are largely based on Ecoinvent (2010). The upstream burdens of concentrate feed production were equated to those for European winter wheat (Ecoinvent, 2010), whilst the burdens for hay were based on those calculated for silage grass production on a dry-matter basis. (Insert Table 4 about here).

Bioenergy scenarios

Eight plausible bioenergy scenarios were developed based on the findings of recent reports (Mistry et al., 2011a; b; Defra, 2011), farm AD visits (Fre-Energy, 2013; Future Biogas, 2013) and feedback from the TWG (2013). Scenarios were parameterised using Farm-adapt to optimise farm operations assuming fixed milk output (Table 5). Key points are summarised below.

- LD-S: All slurry generated on the baseline large dairy farm is digested to produce biogas that is combusted in an on-farm CHP unit.
- LD-SG: Slurry is augmented with a further c.30% dry matter as silage grass diverted from animal feed to improve economic viability (TWG, 2013).
- LD-SMZ: As above, but with fodder maize instead of silage grass used to augment slurry.
- LD-SF: Imported food waste augments slurry, constrained by farm K_2O surplus (K_2O being the first nutrient to reach a surplus following digestate application).

- LD-M: 10% of farm area is dedicated to miscanthus cultivation, supplying biomass heating pellets, to compare the area-based GHG mitigation efficiency of growing crops for AD.
- MD-S: All slurry generated on the baseline medium dairy farm is digested to produce biogas to replace oil-heating only, as a low-cost AD option based on Bywater (2011) and TWG (2013).
- BAD-SGMZ: Based on Defra (2013) and Spackman (2011), the medium-sized dairy farm converts to a beef-plus-AD enterprise supporting 40 beef suckler cows and producing 904.5 tonnes of maize plus 1836 tonnes of grass annually to feed a 112 kWe capacity AD unit. This scenario reflects the current trend of dairy farm consolidation and intensification ((DairyCo, 2013a; TWG, 2013), and is explained further in S4.

Direct LUC effects incurred in the scenarios, shown in Table 5, were calculated as shown in S3. (Insert Table 5 about here). Scenarios involving cultivation of bioenergy crops lead to reduced availability of fodder feed for animals. Farm-adapt was used to calculate changes in on-farm grass and maize production, and quantities of hay and concentrate feed imported, according to economic optimisation under the constraint of constant milk production (Table 5) – except for the aforementioned BAD-SGMZ scenario. The simplification of constant milk production ensured that the functional unit of baseline food production remained constant at the farm level. Although some milk production could be displaced in the bioenergy crop scenarios, it would likely end up on intensive dairy farms using a high proportion of concentrate feed according to the trend for dairy consolidation (DairyCo, 2013a), with similar net consequences for hay and concentrate feed demand to those displayed in Table 5.

Anaerobic digestion model

The efficiency of and emissions from AD units are strongly influenced by design and management factors. Farm-scale AD units are less likely to have covered storage than larger AD units, potentially leading to high NH_3 losses from the elevated TAN content and pH of digestate relative to slurry, and high CH_4 losses (TWG, 2013). Five design and management options were modelled including a range of possible digestate storage infrastructures ranging from “best case” to “worst case” (Table 6), providing the basis for sensitivity analyses via LCAD tool scenario permutations. The middle default option represents the most likely outcome (TWG, 2013): i.e. use of the same open slurry tank as assumed for the baseline farms, but with a higher TAN EF because a crust is unlikely to form on the separated liquid digestate (Misselbrook et al., 2012). The worst case option represents lagoon storage with a TAN EF of 52% (Misselbrook et al., 2012). The 10% default open tank EF for TAN based on Misselbrook et al. (2012) is similar to the 12% average value for Swiss farm biogas units referred to in BFE (2011). The 5% loss factor for CH_4 during open digestate storage is based on Jungbluth et al. (2007). Further details on AD and CHP parameters are included in S6. (Insert Table 6 about here).

Feedstock and digestate characteristics that influence on AD unit efficiency and life cycle emissions are described in supplementary material (Table S.1). It was assumed that digestate was separated (2% DM in the liquid fraction), with P_2O_5 partitioned in the solid fraction (for model simplification) and spread in proportion to requirements amongst the crops. Soil emissions from separated digestate application were calculated using MANNER-NPK assuming use of a trailing shoe. Total N and TAN in the digestate returned to fields after storage were calculated based on feedstock and digestate TAN contents minus fermentation and digestate storage losses (Table 6). Thus, the farm LCA includes the following N cycle for animal manure N (used to calculate fertiliser replacement value of digestate):

$$N_{cad} = TAN_{ex}AH – N_{gas}AH + TAN_{D} – N_{gas}CHP – N_{gas}DS – N_{gas}F – N_{leach}F$$

where N_{cad} is crop available N in digestate, TAN_{ex} is excreted TAN, N_{gas} is gaseous emission of N, N_{leach} is leaching emission of N, AH is animal housing, CHP is combined heat and power combustion, TAN_{D} is digestion increase following digestion, DS is digestate storage, F is field (the latter two terms calculated by MANNER NPK, as described previously). Mineral fertiliser applications were calculated for each crop area based on the nutrient requirements in Table S1.2 minus available nutrients applied in slurry/digestate, to maintain a consistent nutrient cycle.

Expanded boundary effects

Environmental burdens arising from marginal processes are included in Table 4. Marginal electricity displaced by AD-generated electricity is assumed to be generated in combined cycle gas turbine (CCGT) power stations operating at 50% conversion efficiency (DECC, 2012). Under default LCAD settings, AD heat is used to replace oil heating in animal housing and the farm house. LCAD users can specify up to 100% use of the remaining “waste” heat from the AD unit after parasitic requirements, to replace local heat demand – assumed to be oil heating. Miscanthus pellets were also assumed to displace oil heating, incurring processing, transport and combustion burdens described in supplementary material S7. Oil heating is common in rural areas, so was selected as the marginal heat source.

Counterfactual landfill with biogas energy recovery and in-vessel composting operations were modelled in the LCAD tool (S7), on the basis that approximately 60% of food waste in the UK goes to landfill, and most of the remaining 40% to composting facilities (Mistry, 2011a). The proportion of food waste going to landfill is likely to decline rapidly in response to economic and regulatory drivers being implemented under the Waste Framework Directive (2008/98/EC). Farm anaerobic digestion also requires separated organic waste fractions, which are less likely to go to landfill than unsorted municipal waste. Therefore, composting was selected as the most appropriate default (near-term) counterfactual waste management option in the LCAD tool.

Marginal animal feed production incurred in the bioenergy crop scenarios is likely to involve both intensification and agricultural land expansion in various countries (Schmidt, 2008; Mulligan et al., 2010). This study did not attempt to precisely define marginal feed mixes. Instead, simplified indicative marginal feed scenario permutations were generated based on LCAD tool options in Table 2 to illustrate possible consequences. The best case scenario permutation involves all marginal feed supplied by SBME from Argentina and Brazil in ratios proportional to FAO export statistics (FAO Stat, 2013) with no iLUC incurred, and soy oil replacement of palm oil (Schmidt and Weidema, 2008) – detailed in S2. The worst case permutation involves all marginal feed supplied by SBME incurring iLUC, with soy oil replacing palm oil that has incurred iLUC (see S2). The intermediate permutations assume marginal feed is winter wheat, with an iLUC burden calculated for conversion of UK grassland to annual cropping (see S5). SBME substitutes wheat at a ratio of 0.83 to 1.00 on a mass basis according to energy content (DairyCo, 2013b). The GWP and EP burdens arising from iLUC are calculated as per S2 for SBME, and S3 and S5 for winter wheat. Marginal hay production is most likely to occur via intensification of existing grassland (over 64% of UK agricultural area is grassland, much of it extensively managed: FAO, 2013), for which no soil organic C (SOC) change is assumed (IPCC, 2006). Marginal hay production thus incurs the same upstream burdens as hay imported to the baseline farms (Table 4), driven by fertiliser and energy inputs.

Data presentation

The LCAD tool was systematically run for all scenarios under default settings (Table 2), and across a range of permutations, such as 0% and 100% iLUC for marginal concentrate feed production, and best- and worst-case design and management of the AD unit. Scenario permutations were chosen to: (i) generate simple narratives that offer clear insight into risks and opportunities of bioenergy options on dairy farms; (ii) bound possible outcomes according to best- and worst- case possibilities.

Outputs from the LCAD tool were inputted into an MS Excel database. Marginal environmental effects attributable to co-digestion of non-slurry AD feedstocks were calculated by subtracting effects attributable to slurry-only AD from relevant scenario results. Bioenergy crop effects could then be compared using the following reference flows: Mg feedstock digested and hectares of land used for cultivation. The background data sheets in the LCAD tool enabled ALCA calculations to be undertaken for the various farm and AD unit outputs based on system separation and energy allocation across co-products. In order to generate some of the results contained in this paper, and for comparison with CLCA results, ALCA burdens were calculated per L milk or kg beef live weight produced, and per hectare of land appropriated for bioenergy crops.

Results

Slurry-only AD compared with baseline slurry storage

Introducing slurry-only AD to the large dairy farm led to GWP and RDP burden reductions of 14% and 67%, respectively, relative to the default farm baseline with tank slurry storage (Figure 2; Table 7; Table S9.3), but increases of 10% and 9% for EP and AP burdens owing to NH_3 emissions from the AD unit (digestate storage) exceeding avoided NH_3 emissions from slurry storage (Figure 2; Tables S9.1 and S9.2). Environmental effects attributable to the introduction of AD were dominated by changes in dairy operations and AD unit emissions; avoided oil heating and electricity generation made a significant contribution only to RDP (Figure 2). The enhanced TAN content in AD digestate combined with a shift from splash-plate-slurry to trailing-shoe-digestate application led to a 28% reduction in the AP burden from soil emissions (Table S9.2) and small reductions in GWP (Table 7), RDP (Table S9.3) and EP (Table S9.1) burdens arising from fertiliser manufacture and soil emissions. (Insert Table 7 and Figure 2 about here).

The effects of introducing an AD system to the large dairy farm are highly dependent on how slurry is managed in the baseline situation. If the baseline farm has a lagoon slurry storage system, introducing slurry-only AD can reduce GWP by 39%, and also leads to EP and AP burden reductions of 18% and 36%, respectively (Figure 2). Slurry-only AD on the medium dairy farm leads to smaller environmental effects relative to the baseline situation owing to a lower share of animal excreta collected indoors (183 grazing days per year) and use of biogas for heating only (Table 7; Tables S9.1 to S9.3).

Bioenergy scenario comparisons

Under default assumptions and excluding iLUC, all the AD scenarios led to GWP reductions against respective baseline farms, from 2% for the beef-plus-AD scenario relative to the medium dairy farm up to 25% for slurry-plus-food-waste AD on the large dairy farm (Table 7). Co-digestion of grass or maize did not lead to further GWP reductions compared with slurry-only AD on the large dairy farm. Digestion of grass and maize in the BAD-SGMZ scenario did lead to a minor GWP reduction due to the GWP burden of compensatory milk production on the large dairy farm being lower than the GWP burden of displaced milk production on the baseline medium dairy farm – assuming marginal feed required on the large dairy farm did not incur LUC. All AD scenarios led to EP and AP burden increases (Tables S9.1 and S9.2), by up to 63% for AP in the beef-plus-AD scenario, largely attributable to NH_3 emissions from digestate storage and increased crop production to support animals and AD plants in the AD-PGC scenarios. All AD scenarios led to large net reductions in RDP, by between 8% for heat-only AD on the medium dairy and 229% for the beef-plus-AD scenario (Table S9.3).

The absolute ranges of environmental effects calculated for different combinations of scenario permutations (Figures S9.2 to S9.4) emphasise the high sensitivity of CLCA results to assumptions regarding marginal feed type, iLUC and counterfactual waste management for dairy farm bioenergy options. Nonetheless, the rankings of different feedstock options remain similar across impact categories, and waste feedstocks (slurry and food waste) do not lead to GWP increases even under the worst combinations of scenario permutations.

AD design and management influence

Under the default assumption of open tank digestate storage, AD unit operation makes modest contributions to GWP (Table 7) and EP (Table S9.1) burden changes, and large contributions to AP burden changes (Table S9.2). Consequently, variation in digestate storage for different AD unit designs between best- (gas-tight) and worst- (lagoon) scenario permutations had a major impact on all four burdens for all of the scenarios (Figure 3). AP burden changes were particularly sensitive to digestate storage assumptions: for slurry-plus-food waste AD, the AP burden change relative to the baseline farm ranged from –9% for best-case AD design and management to +123% for worst-case AD design and management. Best-case AD unit design and management restricted the EP burden increases for all the AD scenarios to less than 28% above the baseline farm level, and AP burden increases to less than 7%, apart from the beef-plus-AD scenario. GWP change for the beef-plus-AD scenario ranged from –37% for best case to +38% for worst case AD unit design and management (Figure 3). Locating on-farm AD adjacent to a source of heat demand can significantly improve environmental performance in terms of GWP and RDP if all "waste" AD-CHP heat can be utilised (Table S9.4). (Insert Figure 3 about here).

Feedstock comparison

Increased concentrate feed demand led to large dairy operation EP and AP burden increases (fertiliser application for winter wheat feed) and iLUC burdens for crop feedstocks (Figure 4; see Tables S9.1 to 9.2 for breakdown). Food waste AD led to large a large GWP reduction from avoided landfilling, but large EP and AP increases from storage and soil application of food-based digestate. Avoided counterfactual landfill gas electricity generation offset some of the RDP reduction achieved from electricity generation from food waste AD. Feedstock performance also depends on specific farm circumstances. For example, optimised integration of miscanthus on the large dairy farm led to replacement of maize cultivation with an associated SOC sequestration effect (negative GHG emissions for cultivation in Figure 4). (Insert Figure 4 about here).

Food waste AD achieved the greatest net GWP reduction per Mg feedstock DM under default settings, followed by miscanthus pellet heating (Table 8). Grass and maize feedstocks lead to net GWP increases on the large dairy farm, but minor reductions on the beef-plus-AD farm (Table 8), owing to more efficient production of displaced milk output (Figure 4). However, all AD feedstocks lead to net GWP reductions under best-case AD design and management, and achieve a similar improvement when all net AD heat output displaces oil heating, although miscanthus maintains an advantage compared with AD-PGC (Table 8). Table S9.4 summarises other burden effects arising from use of all net AD heat output, whilst Table S9.5 shows the maximum possible GHG mitigation potential of AD-PGC on an area basis, excluding iLUC. (Insert Table 8 about here).

Marginal animal feed effects

Attributional LCA of grass and maize co-digested for electricity generation shows that these feedstocks cannot achieve GHG mitigation under default assumptions such as 5% CH_4 leakage from open tank digestate storage, but can achieve GHG mitigation of up to 5 Mg CO_2e yr^{-1} per hectare cultivated (LD-SMZ) under best case AD design and management (Figure 5). Compared with ALCA, farm level LCA identified higher GHG emissions per hectare of maize cultivated on the large dairy owing to marginal imported wheat feed having a higher GWP burden than fodder maize, while CLCA identified a small net GHG mitigation effect per hectare maize cultivated on the beef-plus-AD farm owing to the aforementioned lower GWP burden of displaced milk production. However, when iLUC was attributed to the cultivation of marginal wheat feed, conversion of the medium dairy farm to a beef-plus-AD enterprise led to a net GWP burden increase of 6.66 Mg CO_2e yr^{-1} per hectare of grass and maize cultivated for AD (Figure 5), leading to an overall GWP burden increase of 38% for the BAD-SGMZ scenario (Table 7). Attributing iLUC to marginal wheat feed also resulted in large GWP burden increases for grass and maize co-digested on the large dairy farm AD unit, equating to 9.2 and 11.8Mg CO_2e ha^{-1} yr^{-1}, respectively (Figure 5) – translating into 915 and 983 kg CO_2e Mg^{-1} DM co-digested, respectively. Nonetheless, the LD-SG and LD-SMZ scenarios still led to overall net GHG emission reductions of 5% relative the farm baseline owing to the predominant effect of slurry co-digestion (Table 7). Miscanthus for pellet heating achieved a net GWP reduction of more than 7 Mg CO_2e ha^{-1} yr^{-1} after accounting for iLUC possibly incurred by cultivation of winter wheat marginal feed. (Insert Figure 5 about here).

In the worst case scenario permutation of all marginal feed comprising SBME from land recently converted from forest and grassland to cropland in South America, cultivation of all bioenergy crops on the dairy farms led to GWP increases exceeding 26 Mg CO_2e ha^{-1} yr^{-1} (up to 4099 kg CO_2e Mg^{-1} DM co-digested maize) (Figure 5). These effects can also be related to bioenergy output. If the net GWP effect excluding conventional energy substitution for each AD scenario is attributed to net bioenergy generated, electricity GWP burdens reach up to 4.26 kg CO_2e kWh_e^{-1} for the BAD-SGMZ scenario where compensatory milk production is based on maize and SBME incurring iLUC (Table S9.6) – compared with a reference GWP for marginal grid electricity of 0.42 kg CO_2e kWh_e^{-1}.

Discussion

Methodological approach

Introducing AD to a dairy farm fundamentally alters material flows through the farm system. In this paper, we have demonstrated the importance of using detailed farm models to comprehensively assess these changes, which are more important than fossil energy substitution for most environmental effects. Particular care was taken to account for nutrient flows and to apply a consistent methodology across bioenergy and counterfactual processes based on realistic assumptions (e.g. calculation of slurry- and digestate-storage NH_3 emissions in proportion to TAN, assuming crust for slurry but not for liquid digestate).

Expanding LCA boundaries revealed a plethora of counterfactual situations against which dairy farm bioenergy options could be compared. Whilst there is an irrefutable logic behind the use of economic models to identify specific marginal effects within CLCA (Weidema, 2001; Earles et al., 2012), this can result in a lack of transparency, mask large underlying uncertainties linked to factors such as volatile commodity prices, and obscure causal links within the net effect (Zamagni et al., 2012). Although the simplified scenario permutations used as the basis for CLCA in this paper may not individually represent the most likely outcomes for bioenergy options on dairy farms, they provide clear narratives that link particular management practices and marginal consequences with quantified environmental effects. We would argue that the transparency of this approach provides valuable insights for stakeholders into the opportunities and risks associated with the introduction of bioenergy production to dairy farms.

Risks and opportunities of on-farm anaerobic digestion

This study confirms that on-farm AD of slurry and food waste is an effective option to reduce GHG emissions and improve resource efficiency but can lead to increased AP and EP burdens, as reported in recent studies (e.g. Hamelin et al., 2011; Rehl&Müller, 2011; Boulamante et al., 2013; Tufvesson et al., 2013). The trade-off risks of increased eutrophication and acidification can be mitigated through more expensive sealed digestate storage, which may require regulation or incentivisation, given the marginal economic viability of small-scale farm AD (Bywater, 2011; Defra, 2014). Biogas production from bioenergy crops is environmentally detrimental and should be avoided – alternative bioenergy crop options such as miscanthus heating pellets offer much greater GHG mitigation potential. However, to ensure economic viability of on-farm AD units, it may be necessary to co-digest non-slurry feedstock, in order to boost FIT income from electricity and enable continuous operation throughout the year (Defra, 2011). In this context, there may be an argument for limited inclusion of crop feedstocks to justify AD investment and leverage the environmental benefits of slurry digestion. Whilst farm co-digestion of food waste results in very positive GHG and resource depletion effects given current counterfactual waste management in the UK, more advanced waste management options such as mechanical-biological treatment coupled with AD are likely to become more prevalent (Montejo et al., 2013), negating many of the environmental benefits of co-digesting food waste on farms. A preferable route to achieving AD treatment of farm wastes would be the construction of shared AD plants, for example under cooperative arrangements, where economies of scale could make good design and management economically viable.

Carbon leakage via animal feed imports

An important finding of this study is the magnitude of possible international "carbon leakage" that can occur via marginal animal feed supply when fodder feed is diverted to AD, or when on-farm fodder production is displaced by bioenergy crops. Whilst other studies such as Tonnini et al. (2012) and Hamelin et al. (2014) have considered iLUC for biogas feedstocks, and Hortenhuber et al. (2011) have quantified the large GWP burden of South American SBME feed caused by iLUC, no studies have so far linked the latter burden to bioenergy crop feedstocks via fodder production displacement on dairy farms. Similar iLUC effects may be incurred by marginal demand increases for other internationally traded feeds grown at the frontier of global agricultural expansion, especially in South America. Attributing iLUC burdens to all marginal animal feed is a simplified worst case assumption. However, in the context of slowing crop yield increases and net food import to the UK and wider EU (Tukker et al., 2013), along with rapidly growing global food demand (BBSRC, 2013), such an assumption represents a plausible and pertinent scenario that emphasises the risks associated with displacement of dairy fodder feed. Dauber et al. (2012) highlight the scarcity of "spare" land for bioenergy production globally, and projected supply constraints underpin the current focus on "sustainable intensification" in agricultural policy (FCRN, 2013). Understanding the routes and magnitudes of carbon leakage effects is critical to ensure that bioenergy policy, and agricultural policy more widely, is adequately designed to minimise such risks.

In conclusion, bioenergy generated from AD of waste feedstocks is an effective option for GHG mitigation and more efficient use of resources. Associated air and water pollution risks can be minimised through appropriate management such as sealed digestate storage and trailing shoe or injection application of digestate, which may require regulation. However, the digestion of crops for biogas production on dairy farms is detrimental to the environment, and at best represents an inefficient pathway for GHG mitigation compared with other crop-based bioenergy options such as miscanthus heating pellets. Furthermore, there is a high risk that the displacement of animal fodder by bioenergy crop cultivation on dairy farms incurs land use change by increasing demand for concentrate feeds such as wheat and soybean meal extract, leading to potentially large net GHG emission increases. Thus, whilst co-digestion of crops may be deemed necessary to stimulate investment in beneficial farm-scale AD of slurry, the quantity of such crops used should be strictly limited to avoid potentially large international carbon leakage and other harmful environmental effects.

Supporting information

Supporting information for this article can be found online in the accompanying MS Word file.

Acknowledgements

The authors are grateful to Defra for funding provided to undertake this research under project code AC0410, and to feedback from the stakeholder experts who attended the workshop and provided information. We would also like to thank anonymous reviewers for their constructive suggestions to improve the original manuscript.

References

ADBA (2013). *Projections for the UK AD and biogas market in 2013/14*. Website last accessed June 2013: http://www.adbiogas.co.uk/2013/05/17/whats-in-store-for-the-uk-ad-market-in-201314-3/

Alderman, G., Cottrill, B.R., 1993. *Energy and Protein Requirements of Ruminants. An Advisory Manual Prepared by the AFRC Technical Committee on Responses to Nutrients*. CAB International, Wallingford, UK

BFE (Bundesamt für Energie) (2011). *Life Cycle Assessment of Biogas Production from Different Substrates.* BFE, Schlussbericht Switzerland.

Biograce (2012). Biograce Excel tool version 4b, downloaded from the Biograce website July 2012: www.biograce.net

Biomass Energy Centre (2013). Web portal accessed June 2013: http://www.biomassenergycentre.org.uk/portal/page?_pageid=77,109191&_dad=portal&_schema=PORTAL

Biotechnology and Biological Sciences Research Council (BBSRC) (2013) Global food security. Downloaded September 2013: http://www.foodsecurity.ac.uk/issue/uk.html

Boulamanti, A.K., Maglio, S.D., Giuntoli, J., Agostini, A., 2013. Influence of different practices on biogas sustainability. *Biomass and Bioenergy* 53, 149-161.

Brown, K., Cardenas, L., MacCarthy, J., Murrells, T., Pang, Y., Passant, N., Thistlethwaite, G., Thomson A., Webb, N., et al. (2012). *UK Greenhouse Gas Inventory, 1990 to 2010.*AEA, Didcot. ISBN: 978-0-9565155-8-2.

BSI (2011). *PAS 2050:2011 Specification for the assessment of the life cycle greenhouse gas emissions of goods and services*. London: BSI. ISBN 978 0 580 71382 8.

Bywater, A. (2011). *A Review of Anaerobic Digestion Plants on UK Farms - Barriers, Benefits and Case Studies.* Royal Agricultural Society of England, UK. Downloaded may 2013: http://www.fre-energy.co.uk/pdf/RASE-On-Farm-AD-Review.pdf

CML (2010). Characterisation Factors database available online from Institute of Environmental Sciences (CML), Universiteit Leiden, Leiden, 2010. Downloaded May 2012: http://cml.leiden.edu/software/data-cmlia.html

CFT (Cool Farm Tool) (2012*).* Available to download at the *Cool Farm Tool* homepage. Downloaded June 2012: http://www.coolfarmtool.org/

DairyCo (2013a). *Farming data*. Downloaded September 2013: http://www.dairyco.org.uk/market-information/farming-data/

DairyCo (2013b). Feed ration ingredients webpage, accessed June 2013: http://www.dairyco.org.uk/technical-information/feeding/common-ration-ingredients/

Dalgaard, R., Schmidt, J., Halberg, N., Christensen, P., Thrane, M., Pengue, W.A. (2008): LCA of Soybean Meal. *Int J LCA 13*, 240–254.

Dandres, T., Gaudreault, C., Tirado-Seco, P., and Samson, R. (2011) Assessing non-marginal variations with consequential LCA: Application to European energy sector. *Renewable and Sustainable Energy Reviews* 15, 3121-32.

Dauber, J., Brown, C., Fernando, A.L., Finnan, J., Krasuska, E., Ponitka, J., Styles, D., Thrän, D., Van Groenigen, K.J., Weih, M. (2012). Bioenergy from “surplus” land: environmental and socio-economic implications. *BioRisk* 7, 5–50.

DECC (2012). *Valuation of energy use and greenhouse gas (GHG) emissions.* Department of Energy and Climate Change, UK Government publication downloaded July 2013: https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/68764/122-valuationenergyuseggemissions.pdf.

Defra (2009b). *AC0209: Ruminant nutrition regimes to reduce methane and nitrogen emissions, final project report*. Downloaded September 2013: http://randd.defra.gov.uk/Document.aspx?Document=AC0209_10114_FRP.pdf

Defra (2010). *Fertiliser Manual RB209*. UK: TSO. Retrievable at http://www.defra.gov.uk/publications/2011/03/25/fertiliser-manual-rb209/

Defra (2011). *Wider Impacts of Anaerobic Digestion: Agronomic and Environmental Costs and Benefits.* Unpublished evidence summary and commentary. Defra, London.

Defra (2012). *2012 Guidelines to Defra / DECC's GHG Conversion Factors for Company Reporting.* London: Defra.

Defra (2012). *Monthly farmgate milk prices*. Agricultural Statistics Series.

Defra (2013). Personal communication with Luke Spadavecchia, 30.04.2013.

Defra (2014). *Comparative life cycle assessment of anaerobic digestion.* Project AC0410 webpage. Last accessed January 2014: http://sciencesearch.defra.gov.uk/Default.aspx?Menu=Menu&Module=More&Location=None&Completed=0&ProjectID=18631

De Vries, J.W., Vinken, T.M.W.J., Hamelin, L., De Boer, I.J.M. (2012). Comparing environmental consequences of anaerobic mono- and co-digestion of pig manure to produce bio-energy – a life cycle perspective. *Bioresource Technol* 125, 239–48.

DfT (Department for Transport) (2010). Web archive, last accessed May 2013: http://webarchive.nationalarchives.gov.uk/20101007153548/http://www.dft.gov.uk/pgr/roads/environment/fuel-quality-directive/pdf/fuelquality.pdf

Dieselnet (2013). Webpage, last accessed May 2013: http://www.dieselnet.com/standards/eu/nonroad.php

Del Prado, A., Crosson, P., Olesen, J.E., Rotz, C.A. (2013). Whole-farm models to quantify greenhouse gas emissions and their potential use for linking climate change mitigation and adaptation in temperate grassland ruminant-based farming systems. *Animal* 7: 373–385.
De Vries, J.W., Vinken, T.M.W.J., Hamelin, L., De Boer, I.J.M. (2012). Comparing environmental consequences of anaerobic mono- and co-digestion of pig manure to produce bio-energy – a life cycle perspective. *Bioresource Technol* 125, 239–48.
Duffy, P., Hanley, E., Hyde, B., O'Brien, P., Ponzi, J., Cotter, E., Black, K. (2013). Greenhouse gas emissions 1990 – 2011 reported to the United Nations Framework Convention on Climate Change. Irish Environmental Protection Agency, Dublin.

Earles, J.M., Halog, A., Ince, P., Skog, K. (2012). Integrated economic equilibrium and life cycle assessment modelling for policy-based consequential LCA. *Journal of Industrial Ecology* 17, 375-384.

EC (2009). *Directive 2009/28/EC of the European Parliament and of the Council of 23 April 2009 on the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC*. OJEU: L 140/16.

Ecoinvent (2010). *Ecoinvent database version 2.2*, accessed via SimaPro.

EEA (2012). *Ammonia emissions*, webpage accessed November 2012: http://www.eea.europa.eu/data-and-maps/indicators/eea-32-ammonia-nh3-emissions-1/assessment-1

FAO Stat (2013). Website last accessed June 2013: http://faostat.fao.org/site/339/default.aspx

FCRN (Food Climate Research Network) (2013) *Food security*. Downloaded September 2013: http://www.fcrn.org.uk/research-library/food-security

FIT Ltd (2013). Feed In tariffs. The information site for the new guaranteed payments for renewable electricity in the UK. Accessed July 2013: http://www.fitariffs.co.uk/

FNR (2009) Biogas-Messprogramm II. (2nd Edition), Gülzow, Available from: http://mediathek.fnr.de/broschuren/bioenergie/biogas/biogas-messprogramm-ii-61-biogasanlagen-im-vergleich.html

FNR (2010). *Leitfaden Biogas: Von der Gewinnung zur Nutzung* (5th Edition). Gülzow, 2010. Available to download from: http://mediathek.fnr.de/leitfaden-biogas.html

FNR (2013).*Biogas data webpage*. Last accessed October 2013: Available at: http://mediathek.fnr.de/grafiken/daten-und-fakten/bioenergie/biogas.html

Fre-Energy (2013). Personal communication during farm visit, April 2013.

Future Biogas (2013). Personal communication during visit to Spring Farm, February 2013.

Gibbons, J. M., Ramsden, S. J., & Blake, A. (2006). Modelling uncertainty in greenhouse gas emissions from UK agriculture at the farm level. *Agriculture, Ecosystems & Environment* 112, 347-355.

Hamelin, L., Wesnæs, M., Wenzel, H., Petersen, B.M. (2011). Environmental Consequences of Future Biogas Technologies Based on Separated Slurry. *Environ. Sci. Technol*. 45, 5869–5877.

Hamelin, L., Joergensen, U., Petersen, B. M., Olesen, J. E., Wenzel, H. (2012). Modelling the carbon and nitrogen balances of direct land use changes from energy crops in Denmark: A consequential life cycle inventory. *GCB Bioenergy* 4, 889−907.

Hamelin, L., Naroznova, I., Wenzel, H. (2014). Environmental consequences of different carbon alternatives for increased manure-based biogas. *Applied Energy* 114, 774–782.

Holm-Nielsen, J.B., Al Sead, T., Oleskowicz-Popiel, P. (2009).The future of anaerobic digestion and biogas utilization. *Bioresource Technology* 100: 5478-5484.

Hörtenhuber, S.J., Lindenthal, T., Zollistsch, W. (2011).Reduction of greenhouse gas emissions from feed supply chains by utilising regionally produced protein sources: the case of Austrian dairy production. *Journal of the Science of Food and Agriculture*, 91, 1118-1127.

IPCC (2006). *2006 IPCC Guidelines for National Greenhouse Gas Inventories.* Retrieved from http://www.ipcc-nggip.iges.or.jp/public/2006gl/index.html

ISO (2006a). *ISO 14040: Environmental management — Life cycle assessment — Principles and framework (2nd ed.).* Geneva: ISO.

ISO (2006b). *ISO 14044: Environmental management — Life cycle assessment — Requirements and guidelines.* Geneva: ISO.

Johnes, P., Moss, B., Phillips, G. (1996). The determination of total nitrogen and total phosphorus concentrations in freshwaters from land use, stock headage and population data: testing of a model for use in conservation and water quality management. *Freshwater Biology* 36, 451–473.

Jungbluth N., Chudacoff M., Dauriat A., Dinkel F., Doka G., Faist Emmenegger M., Gnansounou E., Kljun N., Schleiss K., Spielmann M., Stettler C. and Sutter J. (2007). Life Cycle Inventories of Bioenergy. Ecoinvent report No. 17. ESU-services, Uster, CH, retrieved from: www.ecoinvent.org.

Kloverpris, J., Wenzel, H., Nielsen, P. (2008). Life cycle inventory modeling of land use induced by crop consumption. *International Journal of Life Cycle Assessment 13*, 13–21.

Lapola, D., R. Schaldach, J. Alcamo, A. Bondeau, Liebetrau J. et al. (2012). Emissionsanalyse und Quantifizierung vonStoffflüssen durch Biogasanlagen im Hinblick auf die ökologische Bewertung derlandwirtschaftlichen Biogasgewinnung undInventarisierung der deutschenLandwirtschaft. Project Report: FKZ: 22023606.

Marvuglia, A., Benetto, E., Rege, S., Jury, C. (2013). Modelling approaches for Consequential Life Cycle Assessment (C-LCA) of bioenergy: critical review and proposed framework for biogas production. *Renewable and Sustainable Energy Reviews* 25, 768-781.

Mathiesen, B.V., Münster, M., Fruergaard, T. (2009) Uncertainties related to the identification of the marginal energy technology in consequential life cycle assessments. *Journal of Cleaner Production* 17, 1331-8.

Misselbrook, T.H., Gilhespy, S.L., Cardenas, L.M. (Eds.) (2012). *Inventory of Ammonia Emissions from UK Agriculture 2011*. Defra, London.

Mistry, P, Procter, C., Narkeviciute, R., Webb, J., Wilson, L., Metcalfe, P., Solano-Rodriguez, B., Conchie, S., Kiff, B. (2011). *Implementation of AD in E&W Balancing optimal outputs with minimal environmental impacts* (AEAT/ENV/R/3162 April 2011).

Mistry, P, Procter, C., Narkeviciute, R., Webb, J., Wilson, L., Metcalfe, P., Twining, S., Solano-Rodriguez, B. (2011b). *Implementation of AD in England & Wales: Balancing optimal outputs with minimal environmental impacts - Impact of using purpose grown crops* (AEAT/ENV/R/3220, November, 2011).

Montejo, C., Tonini, D., del Carmen Márqueza, M., Astrup, T.F. (2013). Mechanical–biological treatment: Performance and potentials. An LCA of 8 MBT plants including waste characterization. *Journal of Environmental Management* 128, 661–673.

Mulligan, D., Edwards, R., Marelli, L., Scarlat, N., Brandao, M., Monforti-Ferrario, F. (2010). The effects of increased demand for biofuel feedstocks on the world agricultural markets and areas. JRC, Ispra. ISBN 978-92-79-16220-6.

Nicholson, F.A., Bhogal, A., Chadwick, D., Gill, E., Gooday, R.D., Lord, E., Misselbrook, T., Rollett, A.J., Sagoo, E., Smith, K.A., Thorman, R.E., Williams, J.R., Chambers, B.J. (2013). An enhanced software tool to support better use of manure nutrients: MANNER-NPK. Soil Use and Management (in press). doi: 10.1111/sum.12078

Rehl, T., Müller, J. (2011) Life cycle assessment of biogas digestate processing technologies. *Resources, Conservation and Recycling*, 56, 92–104.

Rehl, T., Lansche, J. Müller, J. (2012).Life cycle assessment of energy generation from biogas—Attributional vs. consequential approach. *Renewable and Sustainable Energy* Reviews 16, 3766– 3775.

Schmidt, J.H. (2008). System delimitation in agricultural consequential LCA – outline of methodology and illustrative case study of wheat in Denmark. *The Int J Life Cycle Assess* 13, 350–64.

Schmidt, J.H., Weidema, B. (2008). Shift in the marginal supply of vegetable oil. *Int J Life Cycle Assess* 13,235–239.

Spackman, P. (2011). *Small-scale AD worth a second look*. Farmers Weekly, 09.09.2011.

Steubing, B., Zah, R., Ludwig, C. (2012). Heat, electricity, or transportation? The optimal use of residual and waste biomass in Europe from an environmental perspective. *Environ. Sci. Technol.* 46, 164−171.

Thomas, C., Young, J.W.O. (1982). Milk from Grass. ICI Agricultural Division, Billingham, UK.

Thomassen, M.A., Dalgaard, R., Heijungs, R., de Boer, I. (2008). Attributional and consequential LCA of milk production. *The International Journal of Life Cycle Assessment* 13:339–349

Tonini, D., Hamelin, L., Wenzel, H., Astrup, T. (2012). Bioenergy Production from Perennial Energy Crops: A Consequential LCA of 12 Bioenergy Scenarios including Land Use Changes. *Environmental Science & Technology* 46, 13521−13530.

Tufvesson, L.M., Lantz, M., Börjesson, P. (2013). Environmental performance of biogas produced from industrial residues including competition with animal feed - life-cycle calculations according to different methodologies and standards. *Journal of Cleaner Production* 53, 214-223.

Tukker, A., Koning, A., Wood, R., Hawkins, T., Lutter, S., Acosta, J., Cantuche, J.M.R., Bouwmeester, M., Oosterhaven, J., Drosdowskih, T., Kuenena, J. (2013). Exiopol – development and illustrative analyses of a detailed global MR EE SUT/IOT. *Economic Systems Research* 25, 50-70.

TWG (Technical Working Group) (2013). Workshop held in Birmingham NEC Hilton Metropole, 20.02.2013.

Vázquez-Rowe, I., Marvuglia, A., Rege, S., Benetto, E. (2014). Applying consequential LCA to support energy policy: land use change effects of bioenergy production. *Science of the Total Environment* 472, 78-89.

Voigt, R. (2008). *Basisdaten zu THG-Bilanzen für Biogas-Prozessketten und Erstellung neuer THG-Bilanzen*. pp.48. Ifeu – Institut für Energie- und Umweltforschung Heidelberg GmbH, Heidelberg.

Warwick HRI (2007). *AC0401: Direct energy use in agriculture: opportunities for reducing fossil fuel inputs.* Warwick HRI.

Webb, J., Misselbrook, T.H. (2004). A mass-flow model of ammonia emissions from UK livestock production. *Atmospheric Environment* 38, 2163–2176.

Withers, P. (2013). Personal communication, April 2013.

WRAP (2010*). Food Waste Chemical Analysis. Chemical characterisation of food wastes collected from Welsh Local Authorities for supporting decisions related to anaerobic digestion process design and operation.* WRAP, UK.

Weidema, B. (2001). Avoiding Co-Product Allocation in Life-Cycle. *Journal of Industrial Ecology* 4: 11-33.

Westhoek, H., Rood, T., van den Berg, M., Janse, J., Nijdam, D., Reudink, M., Stehfest, E. (2011). *The protein puzzle: The consumption and production of meat, dairy and fish in the European Union.* PBL (Netherlands Environmental Assessment Agency), The Hague. ISBN 978-90-78645-61-0

Zamagni A., Guinée J., Heijungs R., Masoni P., and Raggi A. (2012) Lights and shadows in consequential LCA. *The International Journal of Life Cycle Assessment* 17, 904-18.

Figures

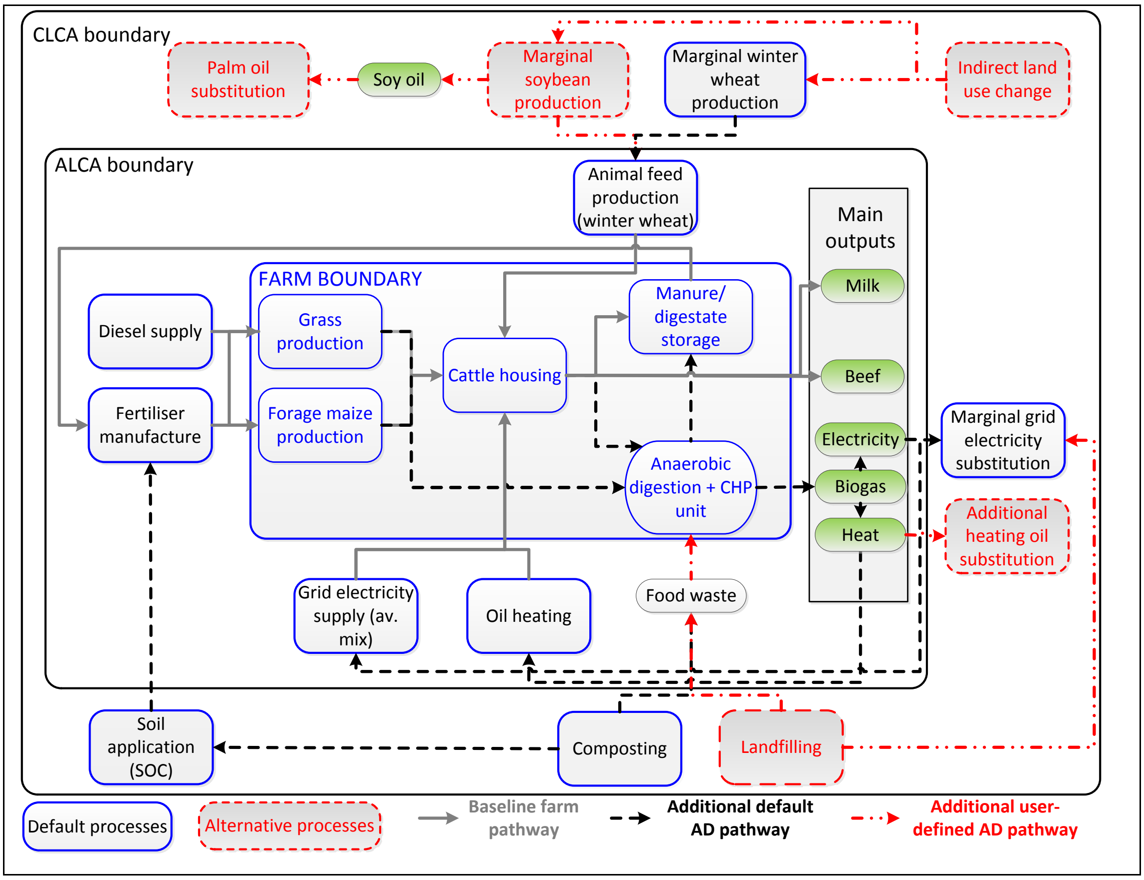

Figure 1. Schematic representation of the main processes and pathways (default and user-defined) considered for the large dairy AD scenarios within the LCAD tool (CLCA boundary)

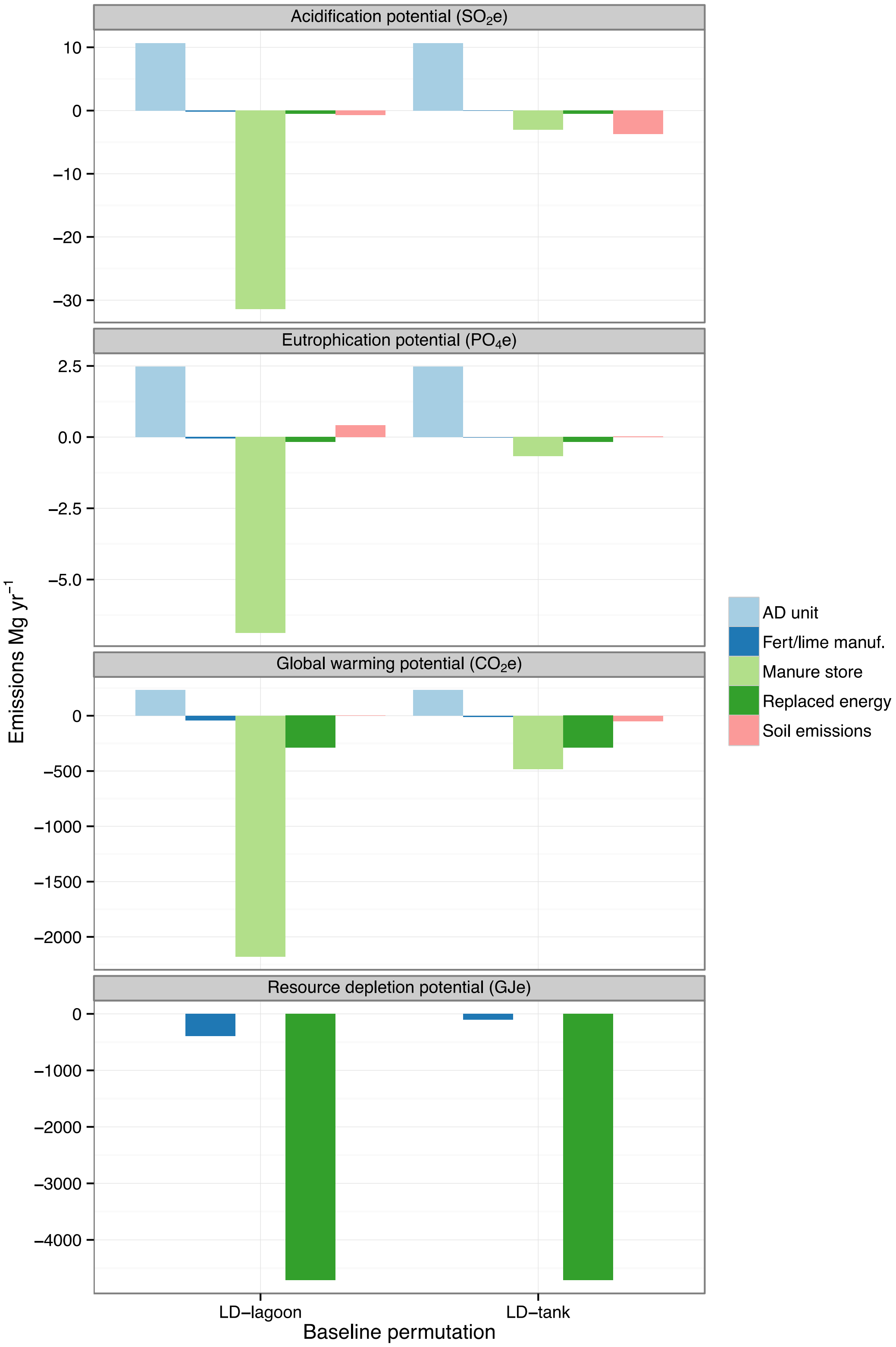

Figure 2. Changes in farm level burdens for the LD-S (slurry-only AD) scenario under default settings (open tank digestate storage), compared with the tank slurry storage default baseline (LD-tank) and lagoon slurry storage alternative baseline (LD-lagoon).

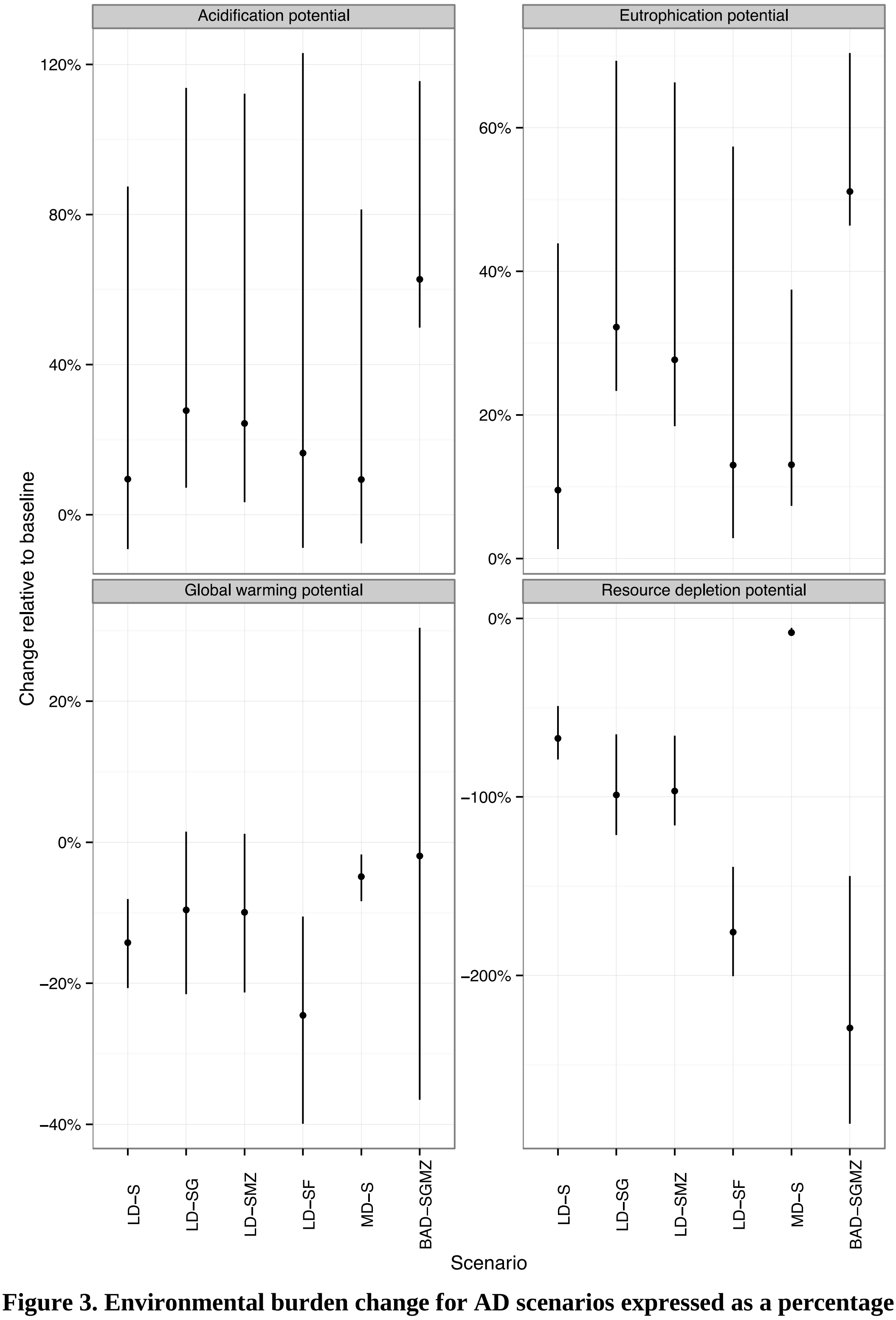

Figure 3. Environmental burden change for AD scenarios expressed as a percentage of relevant baseline farm burdens (LD-BL or MD-BL). Ranges represent AD design and management options: the central bar is the default including open tank digestate storage, the upper bar the worst case including open lagoon storage of digestate and the lower bar the best case including gas-tight storage of digestate (as shown in Table 6).

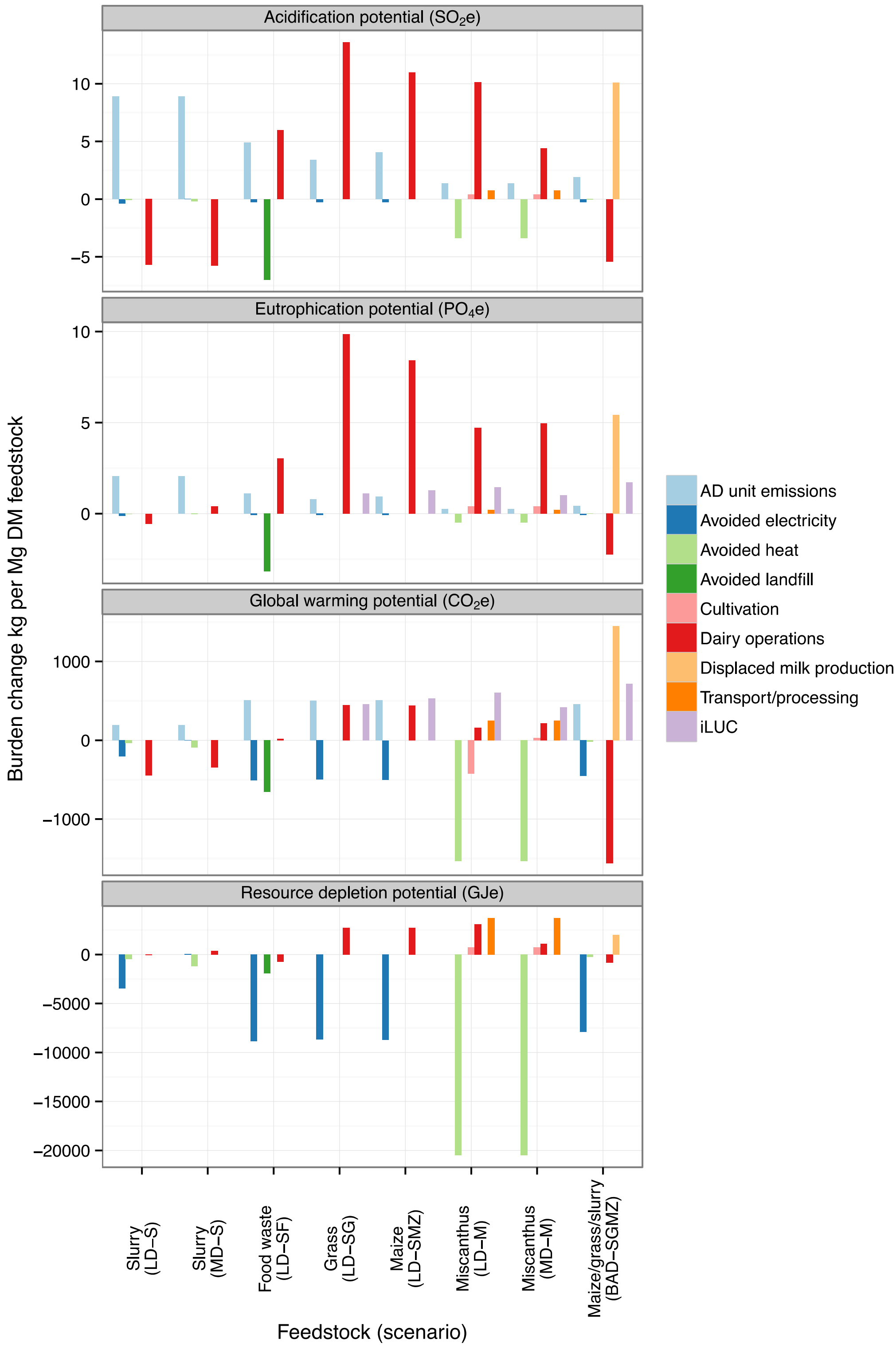

Figure 4. Marginal environmental burden changes arising from one Mg (DM basis) of feedstock digested (slurry), or co-digested (food waste, grass, maize), or used for pellet heating (miscanthus), within large (LD) and medium (MD) dairy farm scenarios. "Dairy operations" includes imported animal feed supply.

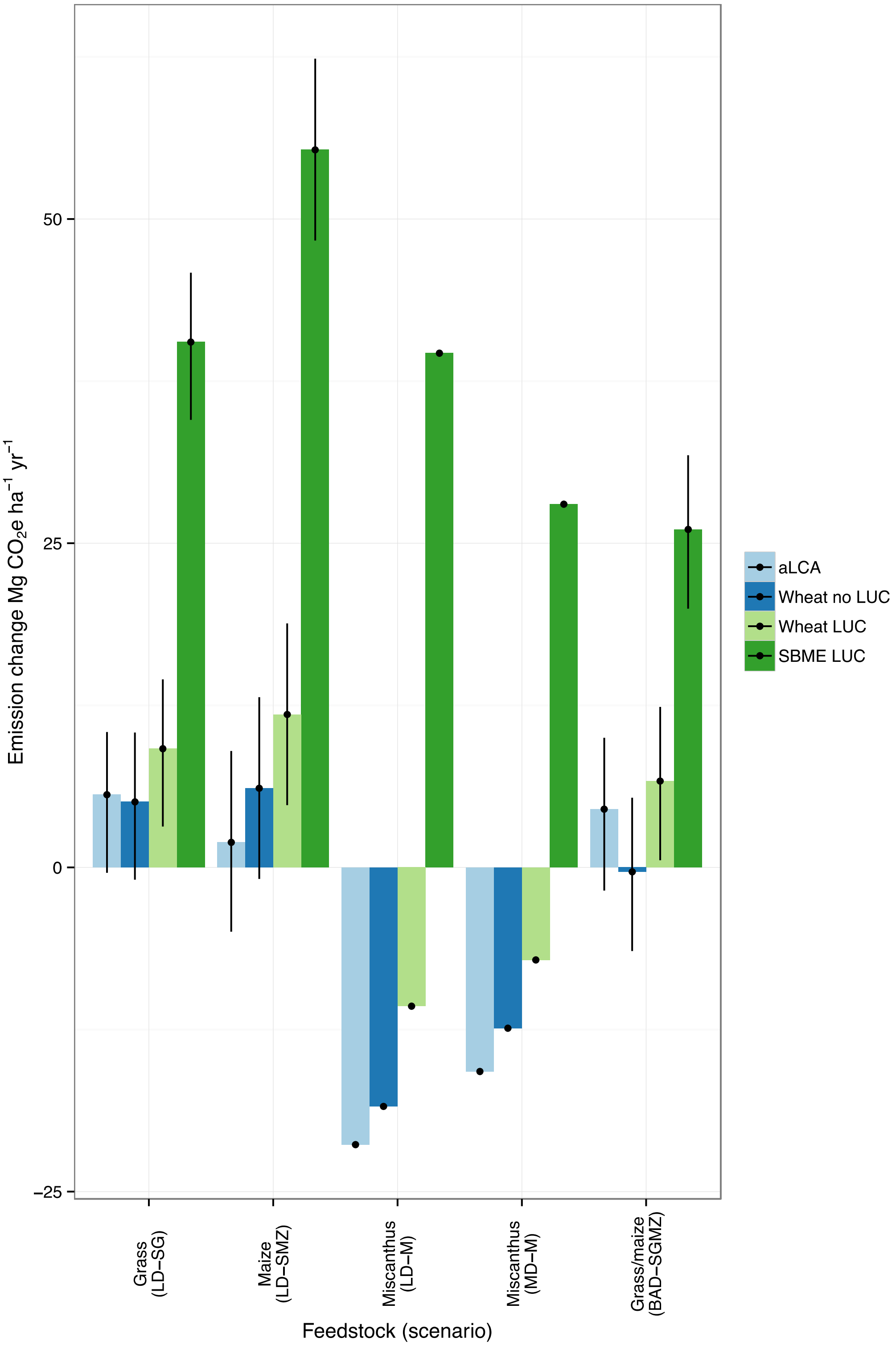

Figure 5. GHG emission change per hectare of bioenergy feedstock cultivation for relevant scenarios, based on attributional LCA, and consequential LCA considering different marginal feed types with and without iLUC (error bars represent range of AD unit design and management performance).

Table 1. Environmental burden characterisation factors applied to emissions and resource use

Impact category	Abbreviation	Indicator	Characterisation factors (per kg)
Global warming potential	GWP	CO_2e	CO_2 1; N_2O 298; CH_4 25
Eutrophication potential	EP	PO_4e	NO_3 0.1; P 3.06; NH_3 0.35; NO_x 0.13; N 0.42
Acidification potential	AP	SO_2e	NH_3 1.6; NO_x 0.5; SO_x 1.2
Resource depletion potential	RDP	MJe	Hard coal 27.91; soft coal 13.96; natural gas (m^3) 38.84; crude oil 41.87.

Table 2. Options available within the LCAD tool to define scenario permutations

Baseline farm slurry storage	Baseline farm slurry application	AD design and management (see Table 8)	Excess* AD heat output utilised	Digestate application method	Marginal feed type	Marginal feed production incurring iLUC	Food waste counterfactual management
Tank storage	**Splash plate**	**Default**	**0%**	**Trailing shoe**	**Winter wheat**	0%	**Composting**
Lagoon storage	Trailing shoe	Good default	50%	Splash plate	Soybean meal extract	25%	Landfilling
		Best case	100%			50%	
		Poor default				100%	
		Worst case					

*Remaining available AD heat output after farm and farmhouse heating supplied
Default permutations in bold

Table 3. Direct emission factors applied across baseline and bioenergy scenarios

Process	Unit	CO_2	CH_4	N_2O-N	NH_3-N	NO_x	NO_3-N	P
Enteric fermentation	Dietary energy as CH_4		[1]0.065					
Housing dairy cows	g NH_3-N LU^{-1} day^{-1}				[2]0.0343			
Housing calves	g NH_3-N $calf^{-1}$ day^{-1}				[2]0.0096			
Manure storage (tank/lagoon)	MCF and TAN EF		[1]0.11/0.68	[2]0.005/0	[2]0.05/0.515			
Fertiliser-N application	Fraction N			[1]0.01	[2]0.018		[3]0.1	
Crop residue N application	Fraction TN			[1]0.01			[3]0.1	
Manure-/digestate- application	Fraction TN			[1]0.01	[4]0.08 – 0.27		[4]0 – 0.28	
Grazing excreta	Fraction TN			[1]0.02	[2,5]0.036		[3]0.1	
All P amendments	Fraction P							[6]0.03
Lime application	kg per kg lime	[1]0.44						
Tractor diesel combustion	kg per kg diesel	[7]3.05	[7]0.000044	[7]0.000048		[8]0.004		

[1]IPCC (2006); [2]Misselbrook et al. (2012); [3]Duffy et al. (2013); [4]MANNER-NPK outputs (ADAS, 2013); [5]Webb and Misselbrook (2004); [6]Withers, pers. comm. (2013); [7]DEFRA (2012); [8] Dieselnet (2013).

Table 4. Environmental burdens from upstream processes

Input	Reference unit	GWP kg CO_2e	EP kg PO_4e	AP kg SO_2e	RDP MJe
Sources of livestock feed/fodder					
Wheat feed	kg	0.577	0.0071	0.0041	3.04
Marginal wheat feed incurring iLUC	kg	1.38	0.0090	NA	NA
Hay/marginal hay	kg	0.528	0.0043	0.0086	2.39
SBME	kg	0.086	0.0039	0.0018	6.82
Marginal SBME incurring iLUC	kg	8.08	0.0103	0.0018	6.82
Fertilizers and other agrochemicals					
Ammonium nitrate-N	kg N	6.10	0.0068	0.024	55.7
Triple superphosphate	kg P_2O_5	2.02	0.045	0.037	28.3
Potassium chloride K_2O	kg K_2O	0.50	0.00077	0.0017	8.32
Lime	kg $CaCO_3$	2.04	0.00040	0.00068	3.31
Crop protection products	kg active ingredient	10.1	0.033	0.097	174
Sources of fuel/energy					
Diesel (upstream)	Kg	0.69	0.00089	0.0062	51.6
Consumed electricity	kWh_e	0.59	0.0076	0.0021	9.48
Marginal electricity generated	kWh_e	0.42	0.000064	0.000226	7.32
Oil heating	kWh_{th}	0.34	0.00011	0.00075	4.55
Transport	tkm	0.081	0.000067	0.0003	1.06

Data based on Ecoinvent (2010), DEFRA (2012), CFT (2012) and own calculations for LUC (see S3).

Table 5. Key features of the two dairy farm baseline and eight tested dairy farm bioenergy scenarios

[1]Scenario name	[2]Annual bioenergy feedstock quantities	CHP capacity	Net electricity output	Utilised heat output	Milk output (L yr^{-1})	Beef output (kg/yr live weight)	Bioenergy feedstock area	On-farm land use change	Concentrate feed/hay imported
	Mg yr^{-1}	kWe	GJ yr^{-1}	GJ yr^{-1}			ha	ha	Mg yr^{-1}
LD-BL	NA	NA	NA	NA	4149102	9242	0	0	972.4/18.90
LD-S	12,016 slurry	72	517	124	4149102	9242	0	0	972.4/18.90
LD-SG	12,628 slurry, 1536 grass silage	135	971	124	4149102	9242	38	9.6 maize to grass	1194.0/193.0
LD-SMZ	13,111 slurry, 1177 maize silage	130	938	124	4149102	9242	26	3.2 maize to grass	1208.1/113.7
LD-SF	12,016 slurry, 2600 food waste	185	1334	124	4149102	9242	0	0	972.4/18.90
LD-M	315 miscanthus	NA	NA	1479	4149102	9242	25	25 maize to miscanthus; 10.1 maize to grass	1209.4/91.7
MD-BL	NA		NA	NA	1013548	2446	0	0	167.0/11.7
MD-S	2366 slurry	NA	NA	62	1013548	2446	0	0	167.0/11.7
BAD-SGMZ	168 slurry, 905 maize silage, 1836 grass silage	112	806	40	0	13870	66	0	2.6 / 25.6
MD-M	107 miscanthus	NA	NA	503	1013548	2446	8.5	8.5 grass to miscanthus	222.9/37.4

[1]BL = baseline; LD = large dairy; MD = medium dairy; BAD = beef plus AD; S = slurry; G = grass silage; MZ = maize silage; F = food waste; M = miscanthus.
[2]Slurry = 10% DM; grass silage = 25% DM; maize silage = 30% DM; miscanthus = 100% DM
[3]Net electricity and utilised heat produced under default settings (excludes potentially utilisable CHP heat in excess of farm demand)

Table 6. Fixed parameters applied across AD design and management permutations based on digestate storage infrastructure types (further details of these options are presented in S3).

	Best case	Good default	Default	Poor default	Worst case
Storage infrastructure type	Gas-tight	Closed	Open tank	Large open tank	Lagoon
[1]Storage loss CH_4 [% produced]	0	2.5	5	7.5	10
[2]CH_4 loss in CHP [%]	0.1	0.5	0.5	0.5	1
[3]Storage loss NH3 [% NH4 in digestate]	0.0	2.0	10.0	16.0	52.0
[2]Mass loss during ensilage [%]	5	10	10	10	25

[1]Derived from Jungbluth et al. (2007) and TWG (2013); [2] Voigt, 2008, [3]Derived from Misselbrook et al. (2012).

Table 7. Contribution analyses for GWP loading (kg CO_2e yr^{-1}) across baseline farms and bioenergy scenarios under default assumptions with and without indirect land use change.

	LD-BL	**LD-S**	**LD-SG**	**LD-SMZ**	**LD-SF**	**LD-M**	**MD-BL**	**MD-S**	**BAD-SGMZ**	**MD-M**
	GWP loading (kg CO_2e yr^{-1})									
Imported feed	571051	571051	790806	757086	571051	746225	102534	102534	15012	148354
Enteric fermentation	2161885	2161885	2124574	2121870	2161885	2109347	576106	576106	141841	576893
Housing/manure store	518469	37961	38570	38562	37961	526265	92697	9945	2393	94750
Electricity use	127720	127720	127720	127720	127720	127720	33254	33191	0	33254
Diesel use	87491	87491	95804	93706	93971	89863	32879	33563	41493	32770
Fertiliser/lime manufacture	154512	145210	135772	146455	90280	162852	101636	105778	122293	93652
Chemical/seed manufacture	8244	8244	7963	8151	8244	7967	1805	1805	1796	2060
Soil emissions	545270	497858	562271	538692	558436	567386	224755	220573	207253	223916
Displaced production	0	0	0	0	0	0	0	0	621103	0
Direct LUC	0	0	-54138	-18046	0	-224066	0	0	0	-8103
(Indirect LUC)	0	0	157336	167347	0	168270	0	0	462189	39689
Transport/processing	0	0	0	0	0	79042	0	0	0	26874
AD unit/combustion	0	230749	423569	410324	574157	0	0	45436	341538	0
Replaced energy	0	-287970	-478504	-464646	-631050	-482716	0	-20115	-351785	-164124
Waste disposal	0	0	0	0	-442941	0	0	0	0	0
Total without iLUC	4174643	3580199	3774410	3759875	3149714	3709885	1165667	1108816	1142939	1060296
Total with iLUC	NA	NA	3931746	3927222	NA	3878155	NA	NA	1605128	1099985
Net effect excluding iLUC	NA	-594444	-400233	-414768	-1024928	-464758	NA	-56851	-22728	-105371
% change excluding iLUC	NA	-14%	-10%	-10%	-25%	-11%	NA	-5%	-2%	-10%
Net effect including iLUC	NA	-594444	-242897	-247421	-1024928	-296488	NA	-56851	439460	-65682
% change including iLUC	NA	-14%	-6%	-6%	-25%	-7%	NA	-5%	38%	-6%

Table 8. Marginal GHG emission change per Mg DM for each AD feedstock (and scenario) under different scenario permutations, excluding possible indirect land use change effects

Scenario permutation	Slurry (LD-S)	Slurry (MD-S)	Food waste (LD-SF)	Grass (LD-SG)	Maize (LD-SMZ)	Miscan. (LD-M)	Miscan. (MD-M)	Maize, grass, slurry (BAD-SGMZ)
	kg CO_2e Mg^{-1} feedstock (DM basis)							
Default (excluding iLUC)	-495	-240	-1971	506	509	-1475	-984	-30
Best case AD design and management	-718	-412	-3799	-95	-74	-1475	-984	-570
100% heat used	-611	-302	-3356	92	93	-1475	-984	-389